2003

Mathematical modeling analysis of Floating Bead Biofilter applications to domestic wastewater treatment

Qiang Wu
Louisiana State University and Agricultural and Mechanical College

MATHEMATICAL MODELING ANALYSIS OF FLOATING BEAD BIOFILTER APPLICATIONS TO DOMESTIC WASTEWATER TREATMENT

A Thesis
Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
In partial fulfillment of the
Requirements for the degree of
Master of Science in Civil Engineering

in

Department of Civil and Environmental Engineering

by

Qiang Wu
B.S., Xiang Tan University, 1985
M.S., Beijing Institute of Light Industry, 1988
August 2003

ACKNOWLEDGEMENTS

I would like to express my appreciation and thank everyone who has encouraged me throughout my graduate career.

My sincere appreciation to Dr. Ronald F. Malone, Chevron Professor of Civil & Environmental Engineering, for his time, guidance, encouragement, wisdom, honesty, cheerfulness, and patience, throughout all the days I worked with him at LSU. He provided me with an opportunity I feel I would otherwise have missed. His contributions and dedication to this research compelled me to work harder.

I offer my sincere appreciation to Dr. John J. Sansalone, Associate Professor of Civil & Environmental Engineering and also my co-advisor, for his time, guidance, encouragement, patience, and support, through most of the days I worked with him at LSU.

My special thanks to Dr. Donald D. Adrian, Rubicon Endowed Professor of Civil & Environmental Engineering, for being on my committee and for his time, encouragement, wisdom, and comments on my thesis and research.

I would like to thank the members of EiEL group: Dr. Rusch, Lance, Dr. Tingzong Guo, Sarah, and Cynthia. I also thank Delta Environmental Equipment, Inc., Board of Regents. I would also like to thank Mrs. Sandra Malone, Dr. Dingfang Liu and Mr. Shelby Stewart for their time and suggestions on my thesis.

To my parents and brothers, thank you for all of your love, solicitude, and encouragement since I came to America.

Finally, I would like to express my whole love and appreciation I have for my wife, Wei Cai. She has supported and encouraged me from the first time I thought about entering graduate school. Without her and my sons, Shang & Edwin, by my side, this research would not have been possible. I love you, Wei, Shang and Edwin.

TABLE OF CONTENTS

LIST OF TABLES

TABLE PAGE

LIST OF FIGURES

ABSTRACT

Floating Bead Biofilters (FBFs) have been applied to aquacultural recirculating tanks and domestic wastewater treatment systems for controlling total ammonia nitrogen (TAN), biochemical oxygen demand (BOD), and total suspended solids (TSS). Support modified media in these FBFs provide a large surface area (1150~1475 m^2/m^3) so that the active biofilm can be retained in the FBF by attaching to the media surface. Understanding the theories involved in biofilm processes greatly helps in sizing, designing, and modeling of FBF systems. Fundamental biofilm processes like mass transport of various substrates into the biofilm and the substrate utilization within the biofilm were studied.

A mathematical model (MSB Model) was set up to predict the development of the FBFs characteristics such as biofilm growth, substrates utilization, dissolved oxygen consumption, BOD loading removal, volumetric oxygen consumption rate by filter (OCF), and bead bed volume under the different conditions. This model was then calibrated with a set of bioclarification data. The model results were consistent with literature defining the relationships between dissolved oxygen consumption, BOD loading removal, and biofilm growth. This model is specifically used to predict design parameters for FBFs in municipal sewage treatment systems. The entire study was based on the following experimental parameters: OCF, dissolved oxygen (DO), hydraulic loading, BOD loading, maximum ratio of BOD removal to OCF (MX-factor).

CHAPTER 1: INTRODUCTION

The growth of industry and the development of towns and cities have resulted in the design, construction and operation of wastewater treatment facilities of increasing size and complexity. However, the greatest number of domestic sewage treatment works in both the industrialized and developing regions of the world are still associated with small communities (towns, camps, coastal homes, villages, and hamlets) with low populations. Recent surveys have shown that over 25% of the residential dwellings in the United States are not served by centralized collection and domestic wastewater treatment systems (Metcalf & Eddy, 1991). Many common coastal conditions present serious constraints on the effectiveness of both central treatment facilities and in-situ treatment systems (septic and extended air systems). As populations grow in such communities, the ability to sustain a healthy coastal environment is becoming increasingly difficult. Coastal states, such as Louisiana, with a large number of small communities, camps, and intermittent-use dwellings have identified the very pressing need for effective, affordable treatment technologies. The research on developing an effective and competitive sewage treatment system will definitely present significant benefits to provide a viable and economic alternative to septic systems, extended air systems, and centralized wastewater treatment. It is very important that such a product is correctly designed, without unnecessary extra capacity, to produce the required effluent quality. Such design must also ensure reliability, flexibility, and safety of operation, together with durability and reliability, while ensuring ease of maintenance and a low use of electrical energy and labor. Equally important, the treatment system should ensure that a public nuisance is not created by its appearance, smell, or noise.

Domestic sewage consists of particles of various sizes suspended in a relatively weak solution of organic and inorganic compounds. It originates from human and other household wastes. This suggests that it should be composed, for the most part, of simple elements of carbon, hydrogen, oxygen, nitrogen, and sulfur, and that its constituent compounds will be mainly carbohydrates, fats, proteins, and urea. An estimate of the relative proportions of carbon, nitrogen, and ammonia in the main components of domestic sewage is given in Table 1.1 (Nicoll, 1988). The figures indicate that although the bulk of the organic nitrogen is derived from excreta, only about 60% of the organic carbon comes from this source. The main components, revealed by the analysis, included carbohydrates, amino, fatty, soluble acids, esters, and sugars, in which glucose and sucrose were predominant. Acetic, propionic, and butyric acids were also found (Painter and Viney, 1959).

Table 1.1: Proportion of Carbon, Nitrogen, and Ammonia in Domestic Sewage (Nicoll, 1988)

Component	Organic Carbon Excluding Urea (%)	Organic Nitrogen (%)	Ammonia Plus Urea (as N) (%)
Feces	46	44	-
Urine	13	50	100
Dish Washing and Food Preparation	22	6	-
Personal and Clothes Washing	19	-	-
Whole Sewage	100	100	100

The floating bead filter (FBF) is an emerging fixed-biofilm technology. The early use of floating bead filters dates back to the mid-1970's when researchers utilized them as biofilters to support high density rearing of food and game fish in Idaho (Cooley, 1979). Adoption of these early air-washed bead filters at other sites was limited. In the

1980's, work performed at Louisiana State University (LSU) demonstrated that a hydraulically washed bead filter was capable of providing solids control (clarification) and biofiltration for a high-density catfish rearing system (Wimberly, 1990). Development of mechanically washed units (Malone, 1992; Malone, 1995), which were compact and simple to operate, overcame many of the operational difficulties experienced by earlier designs. Shortly thereafter, the "bubble-washed" or "hourglass" configuration (Malone, 1993) was developed for use on outdoor ornamental or garden ponds. Since 1989, bead filters have been tested on aquacultural recirculating systems along with a wide variety of specialized applications and display a bioclarification behavior similar to sand filters (Malone & Beecher, 2000). Most recently, the FBFs are being examined for treatment of domestic wastewater and sewage, with particular emphasis on their clarification capabilities.

A FBF functions as a physical filtration device (or clarifier) by removing solids while simultaneously encouraging the growth of bacteria that remove dissolved organic matter from the raw wastewater through biofiltration processes. These granular media are plastic beads, which float in a FBF's bed. This distinguishing feature of the bead bed allows it to be cleaned to release solids and excessive biofloc, while providing large amounts of surface area for the formation of biofilm. Presently, there exist two classes of FBFs. One is the gentle washed category, which includes most hydraulic and air washed units, and another is the aggressively washed category, which includes propeller-washed and paddle-washed filters. The former displays reduced biofilm abrasion during backwashing and must be washed frequently. Conversely, the latter inflicts damage to a relatively heavy biofilm during backwashing.

Efficient FBFs depend on the properties of media most importantly their specific surface area (SSA). Usually, the unmodified polyethylene bead provides high SSA. For example, the unmodified polyethylene bead with a diameter of 2-3 mm used in the FBFs typically has 1150-1475 m^2/m^3 SSA (Malone, 1993). Comparing to unmodified beads, the modified polyethylene beads have higher SSA. Thus, SSA directly affects biofilm performance of FBFs. In addition, shapes and configurations of FBFs vary widely. Different FBFs structures make them to present significantly different characteristics.

CHAPTER 2: OBJECTIVES

The use of FBFs as bioclarification simplifies the process of domestic wastewater treatment. This thesis summarizes what is known about the proper operation and management of floating bead filters.

This research will build upon the development of FBFs for aquaculture wastewater and domestic wastewater treatment to develop a combined mathematical model that can facilitate sizing and support of floating bead biofilters of scale. This model provides quantitative estimates of OCF, BOD removal rate, bead bed volume, bed recycling flowrate, and system hydraulic loading ranges. Such sizing criteria ultimately permit appropriate and rational designs for various FBFs applications subject to various wastewater loadings from domestic and aquacultural sources. The specific objectives are as follows:

1. Critically review the current knowledge of the biological fixed-film reactors, mathematical models, and mass transport phenomena involving microbial population dynamics, biofilm growth, biofiltration and clarification, diffusive substrate mass transport characteristics, and biochemical reaction of soluble and particulate substrates.
2. Use the current knowledge of FBFs processes to mathematically develop a comprehensive FBF model for domestic wastewater treatment applications.
3. Based on the developed mathematical model, use the mathematical model and computing software to calibrate the biofilter kinetic equations, and predict the development of biofilms, BOD removal rate, and OCF variation.

4. Use the mathematical model to quantitatively describe design and operation aspects of FBFs that must be known for practical domestic wastewater treatment demonstration in a full size facility.

CHAPTER 3: LITERATURE REVIEW

It was recognized in the 1880s that the effective purification of sewage required the action of 'biological film' and other organisms under conditions favorable to their propagation. Biological filtration is a traditional step in sewage treatment. Fixed-film processes are characterized by bacteria being attached to a solid surface in the form of a biofilm that removes soluble substrates, colloidal matters and fine suspended solids.

3.1 Monod Kinetics Model

Originally, exponential growth of bacteria was considered to be possible only when all nutrients, including the substrate, were present in high concentration. However, it was found later that bacteria grow exponentially even when one nutrient is present only in limited amount (Monod, 1949). Furthermore, the value of the specific growth rate coefficient, μ, was found to depend on the concentration of that limiting nutrient, which can be the carbon source (the substrate), the dissolved oxygen, nitrogen, or any other factor needed by the organisms for growth. In this thesis, the only situations, where organic substrate or dissolved oxygen are growth limiting, are considered. Typically, the single Monod kinetic equation for the substrate removal rate is expressed as follows:

$$\mu = \frac{\mu_m S}{K_S + S} \tag{3.1}$$

where K_S is the half-saturation coefficient. It determines how rapidly μ approaches μ_m and is defined as the substrate concentration at which μ is equal to half of μ_m. The smaller K_S is, the lower the substrate concentration at which μ approaches μ_m. S is substrate concentration, mg/L.

The Monod equation has been used extensively in the development of models describing the continuous cultivation of microorganisms. Based on previous research, the Monod equation can adequately describe the effect of biodegradable COD or BOD on the specific growth rate of bacteria. Also, the growth of a heterogeneous assemblage can be expressed as ‘biomass’ by the Monod empirical equation (Andrews, 1971; Chiu et al., 1972; Eckhoff and Jenkins, 1967; Gaudy and Gaudy, 1971; McCarty and Lawrence, 1970). The value of μ_m and K_S obtained from mixed culture systems are in reality average values resulting from many interacting species (Chiu, et al., 1972; Gaudy and Gaudy, 1971; Ghosh and Pohland, 1971). Consequently, it has been recommended that μ_m and K_S be characterized by ranges, rather than by single values.

The substrate removal corresponding to the biofilm growth is presented as follows.

$$r_{V,X} = \frac{\mu_m S}{K_S + S} \cdot X_F \tag{3.2}$$

$$r_{V,S} = \frac{r_{V,X}}{Y} \tag{3.3}$$

where $r_{V,S}$ is the substrate volumetric reaction rate, g/m^3-day; $r_{V,X}$ is the biofilm growth rate, g/m^3-day; Y is yield coefficient, g biomass/g substrate mass; X_F is the effective biomass in the system, g/m^3.

The Equation (3.2) usually applies in a situation where only the substrate, S, is a limiting factor for the biofilm growth. Alternatively, μ_m can be seen as the maximum specific growth rate under given set of environmental conditions defined by parameters such as temperature, dissolved oxygen, pH, and other nutrients.

Based on the enzymatic model of Haldane (1930), Andrews (1968) proposed a function to represent the effects of inhibitory organic substrates on microorganism growth rates.

$$\mu = \mu_m \frac{S}{K_S + S + S^2 / K_I} \qquad (3.4)$$

where K_I is the inhibitory coefficient, mg/L. When K_I is very large, the Andrews equation simplifies to the Monod equation, demonstrating that μ_m and K_S have the same meaning in both equations. Unlike the situation for a non-inhibitory substrate, however, μ_m cannot actually be observed and is a hypothetical maximum specific growth rate that would be attained if the substrate were not inhibitory. Furthermore, since μ_m cannot be observed, K_S also takes on a hypothetical value. The most outstanding characteristic is that μ passes through a maximum, μ^{max}, at substrate concentration S^{max}, where

$$\mu^{max} = \frac{\mu_m}{2\sqrt{(K_S / K_I)} + 1} \qquad (3.5)$$

and $$S^{max} = \sqrt{(K_S \cdot K_I)} \qquad (3.6)$$

The Equation (3.5) indicates that the degree of inhibition is determined by (K_S/K_I), and not just by K_I alone. The larger (K_S/K_I), the smaller μ^{max} is relative to μ_m, and thus, the greater the degree of inhibition.

Furthermore, μ^{max} and S^{max} are important in the determination of the kinetic parameters for inhibitory substrates. Equation (3.4) has been used widely in the modeling of various wastewater treatment systems.

3.2 Biofilm Kinetics

Biofilms are very complex, both physically and microbiologically. In fact, they are so complex that it is impossible to fully explore all their aspects based on the wide variety of present clarification and biofiltration (Leslie Grady *et. al.*, 1999). These dense layers of bacteria are characterized by their ability to adhere to a solid medium. They form a fixed film of polymers in which the bacteria are protected against sloughing off (Henze *et. al.*, 1997). Usually, fixed-film biofilters have a short hydraulic retention time (HRT) and a long active fixed-film retention time or high microorganism retention time (MCRT).

The disadvantage of fixed-film reactors is the low transport efficiency of biomass. The substances to be removed must be carried through the biofilm to be removed by the bacteria. This transport takes place by molecular diffusion, which is a slow process (Leslie Grady *et. al.*, 1999).

Steady state biofilm process models describing a soluble substrate and single species are well established. They have been applied successfully to various biofilm reactors (Kissel *et. al.*, 1984; Harremoes, 1978; Characklis and Marshall, 1990; Rittmann and McCarty, 1981; Shieh and Keenan, 1986; Golla and Overcamp, 1990). The physical, chemical and biological processes of biofilm development are transportation and absorption of substrates, attachment of microorganisms to medium surface, and microbial transformations (growth and decay). They result from substrate utilization and biofilm detachment by hydraulic shears stresses (Peyton and Characklis, 1993). These dynamic processes have been investigated by various researchers.

3.2.1 Single-species Biofilm Kinetic Models For Soluble Substrates

Figure 3.1 illustrates a conceptual biofilm model. In the model, a base film is overlaying with an irregular surface film that, in turn, is enclosed by a uniform water boundary layer. Most mathematical models of biofilm systems consider the surface of the biofilm to be negligible, and therefore consider only the base film, carbon oxidation, soluble organic substrate transformation, dissolved oxygen consumption, nitrification, and denitrification. Transport of nutrients and dissolved oxygen to and from the bacteria within the biofilm is normally assumed to be controlled by molecular diffusion alone (Gujer and Wanner, 1990). Transport from the bulk fluid to the biofilm on the other hand is assumed to be dominated by advection or turbulent diffusion (Henze *et. al.*, 1997).

The biofilm grows attached to a solid support medium, which is usually impermeable. In general, the biofilm can be divided into two zones, the base film and the surface film. Both contain an assemblage of microorganisms and other particulate material bound together by a matrix of extracellular polymers (Leslie Grady *et. al.*, 1999). The base film consists of a structured accumulation, with well-defined boundaries. Transport in the base film is controlled by molecular diffusion processes. The surface film provides a transition between the base film and the bulk liquid, with transport within the bulk liquid dominated by advection (Henze *et. al.*, 1997). The relative thickness of the base and surface films depends not only on the hydrodynamic characteristics, substrates concentration, and environmental factors of the system, but also on the nature of the microorganisms in the biofilm. Consequently, one biofilm may have almost no surface film whereas another may be entirely surface film (Characklis and Marshall, 1990; Leslie Grady *et. al.*, 1999). There is normally a relative motion between the biofilm and the bulk

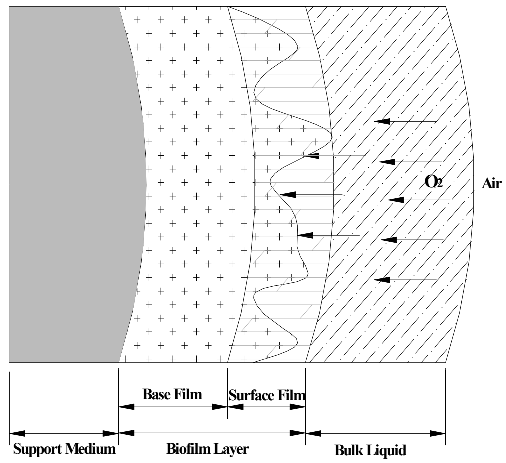

Figure 3.1 Conceptualization of a Biofilm System (After Characklis and Marshall, 1990)

liquid, with the one moving depending upon the configuration of the attached growth process. Mass transfer from the bulk fluid to the biofilm depends on the hydrodynamic regime. Biofilm systems may also contain a gas phase that provides dissolved oxygen.

3.2.2 Mass Balance Analysis

An idealized biofilm is assumed to be homogeneous. The impact of the surface layer is neglected. In the bulk liquid, outside the biofilm, substrate concentration is assumed to be *S*. The transport into the biofilm takes place by molecular diffusion using the diffusion coefficient *D*. For an infinitesimal section of the biofilm, the following mass balance on substrate can be set up under steady state conditions.

$$[\text{Mass}]_{\text{in}} = [\text{Mass}]_{\text{out}} + [\text{Mass}]_{\text{removed}} \quad (3.7)$$

$$M_{out} = M_{in} - \frac{dS \cdot V}{dt} - r_{V,S}\,dx \quad (3.8)$$

As the transport through the cross section exclusively takes place by diffusion, the effective diffusion equation is (Wanner and Gujer, 1986; Rittmann and McCarty, 1980)

$$\frac{1}{D} \cdot \frac{dS}{dt} = \frac{d^2 S}{dx^2} \quad (3.9)$$

$$r_{V,S} = D\frac{d^2 S}{dx^2} \quad (3.10)$$

Equation 3.9 shows that the substrate volumetric reaction rate is proportional to the gradient of the substrate concentration distribution curvature.

Internal diffusion of mass within the biofilm is normally characterized by the Fick's Law (Equation 3.9) (Rittmann and McCarty, 1980). To model the external mass transport of substrates, a diffusional layer surrounding the biofilm is assumed. Then intrinsic substrate utilization, the internal, and the external mass transport are used to estimate the concentration profile. The substrate flux and the apparent volumetric reaction rate can then be estimated for a given bulk substrate concentration (Skowlund and Kirmse, 1989).

Most mathematical models for biofilms assume that dissolved oxygen and other nutrients are transported to the biomass within the biofilm by diffusional processes alone (Rittmann and McCarty, 1980; Skowlund and Kirmse, 1989; Henze *et. al.*, 1997). Other simplifying assumptions for kinetic models usually include (Rittmann and Manem, 1992; Skowlund and Kirmse, 1989; Skowlund, 1990):

1) Biofilm consists of autotrophs and heterotrophs uniformly distributed throughout the film;
2) Model derivation is considered in a single space dimension;
3) Steady state biofilm is considered.

3.2.3 Zero-Order Kinetics Model

For the zero-order reaction, there exist a number of empirical expressions and some simpler equations. Harremoes (1978) developed a biofilm model by approximating substrate utilization rates with zero-order kinetics. The substrate utilization rate follows a zero-order kinetics at high substrate concentrations in which $S >> K_S$, then $\mu \approx \mu_m$. This can be observed from Monod equations given by

$$r_{V,X} = \frac{\mu_{\max} X_B S}{K_S + S} \tag{3.11}$$

which yields the zero-order reaction rate

$$r_{V,X} = \mu_{\max} X_B \tag{3.12}$$

and another equation as follows:

$$r_{V,S} = \frac{kX_B S}{(K_S + S)} \tag{3.13}$$

which yields the zero-order reaction rate

$$r_{V,S} = kX_B \tag{3.14}$$

where X_B is biomass for bead bed in the biofilter, g/day; $k = \frac{\mu_m}{Y}$.

3.2.4 First-Order Kinetics Model

Based on Equation (3.1), the simplification can be made. It becomes another expression if substrate concentration, *S*, is assumed to be much smaller than K_S.

$$\mu \approx \frac{\mu_m}{K_S} S \tag{3.15}$$

This is a first-order reaction expression with respect to the substrate concentration, *S*.

For the first-order reaction, the reaction rate will vary with the substrate concentration. When $K_S >> S$ in the equation (3.11), a first-order reaction rate will occur. Harremoes (1978) set up a biofilm model with first-order rate kinetics. Substrate utilization rates are estimated by using the first-order relations between the removal rate and substrate concentration.

For the general first-order substrate removal rate, the expression is

$$r_{V,X} = \frac{\mu_{\max} X_B S}{K_S} \tag{3.16}$$

According to the Equation (3.9), a mass balance equation for substrate in biofilm with first-order kinetics is derived:

$$r_{V,S} = \frac{kX_B S}{K_S} = D\frac{d^2 S}{dx^2} \tag{3.17}$$

For the biofilm process, the only new parameter, which comes into use, is the diffusion coefficient *D*. The measurement of this coefficient in the biofilm is uncertain and mostly empirical in its value range. In practice, the diffusion coefficient can be put equal to or a little lower than the molecular diffusion coefficient. The Table 3.1 shows that the variation of diffusion coefficient D is wide and depends on the surface structure of the biofilm (Siegrist and Gujer, 1985).

Table 3.1: Diffusion Coefficients for Oxygen and Organic Substrates, Stoichiometric Conditions, and Estimated Removal Rate in Biofilms (Siegrist and Gujer, 1985)

Substance	D 10-4, m^2/day	$Vo_{2,S}$ gCOD/gO_2	k_0 kgCOD/m^3-day
O_2	1.7-2.1	-	25-200
Acetic acid	0.3-0.7	2.1	230-300
Methanol	0.8-4.0	1.2	40-110
Glucose	0.1-0.7	2.4	350-550
unspec. COD	0.3-0.6	1.4-2.0	50-500
unspec. BOD	0.3-0.6	0.8-1.2	25-250

3.3 Multi-species Biofilm Kinetics Models For Soluble Substrates

In the practical biofilm reactors, a layer of fixed-film biomass, which is attached on the surface of support media, is called a multi-species biofilm because it is composed of several different microbial species. A multi-species biofilm is subject to interactions such as symbiosis or competition for space and, in some cases, for common substrates. Its microbial composition (i.e., both the relative abundance and spatial distribution of the species) is mainly determined by three processes that take place within the biofilm: (1) microbial conversion of substrates; (2) volume expansion of biomass; and (3) transport of substrates by molecular diffusion (Wanner and Gujer, 1986).

In the process of multi-substrate utilization, biological removals are almost always redox-processes, which require two substrates: an oxidant and a reductant.

Sometime they are also called electron donor and electron acceptor. The organic substrate is donor and the dissolved oxygen (DO) is acceptor. One of the most significant results of the biofilm kinetics is the determination as to which substrate is limiting for the removal. This depends on the concentration, the removal rate, and the diffusional rate for each of the two substrates.

3.4 Particulate Substrates Models For Fixed-film Reactor

A considerable fraction (60-85%) of the organic matter in a typical sewage system is particulate (Levine, et al., 1985; Ødegaard, 1997). The main mechanisms which contribute to the removal of suspended particles are (i) straining, (ii) interception, (iii) flocculation, (iv) sedimentation, and (v) adhesion to biofilm growth. However, the exact removal mechanisms of particulate organics and their effects on substrate diffusion and biodegradation, oxygen transport, and microbial growth in a floating bead filter are still not fully understood. Few researchers have studied biofilm kinetic model with particulate substrates (Arvin and Harremoes, 1990; Rohold and Harremoes, 1981; Sarner, 1986) and particle transport to the biofilm surface and dissolved oxygen penetration within the biofilm (Bower, 1987).

Ødegaard and Helness (1999) demonstrated that high-rate domestic secondary wastewater treatment could be obtained in a small-scale plant based upon a highly loaded moving-bed biofilm reactor (retention time<30 minutes). In their research, suspended solids removal of 75-85% and COD removal of 60-70%, corresponding to effluent concentrations in the range of 30-35 mg/L SS and 100-125 mg/L COD, can be expected for sewage of normal strength. A good quality effluent can be achieved if a downstream sand filter is included. They also found that the head-loss was low. A typical biofilter-bed depth is 1.0 m and a typical design head-loss would then be 500 mm through the filter-bed itself. The key design parameter is the SS loading (kg SS loading/m^2-hour). At a SS loading of a kg SS/m^2-hour, a filter run time of 10 hours can be expected in a 1.0 m deep filter-bed, while increasing the SS loading rate to 3 kg SS/m^2-hour reduces the filter run time to about 5 hours.

Sarner (1986) found that the loading of fine particulates decreased the efficiency of the removal of dissolved organic matter in a trickling filter process. The actual mechanisms and the interaction between the particulate and dissolved organic matter are not well understood even if one assumes that both forms of organic matter will compete for the limited dissolved oxygen once the particulate organics undergo hydrolysis. In addition to the particulate matter in the influent, the biomass sheared-off from the biofilm surface may also affect the subsequent biofilm processes downstream. The presence of particulate organics in recirculating treatment systems also affects the biofilm performance (growth and decay).

During the last decade, Mouri and Niwa (1993), Okubo et al. (1990), and Tanaka et al. (1981) have been interested in the use of floating-media filtration for high-rate particle separation. Floating-media filtration usually operates with the liquid flowing upwards through a medium-bed of floating filter. Generally, the medium is coarse (2-10 mm), giving a high porosity and consequently a high sludge accumulation capacity and a low head-loss. The floating filters have been operated at high filtration rates (5-50 m/hour). The low head-loss and long filter run times make these coarse filters of interest for various applications both as a roughing primary filter and as an intermediate separation reactor after biological or chemical pretreatment.

Also, Rohold and Harremoes (1981) hypothesize that particulate substrates must be hydrolyzed by extracellular enzymes before they can be transported into the biofilm. However, Ro and Neethling (1991) showed that biofilm was filled with water and for the most part devoid of bacteria. This indicates that fine particulates may actually diffuse into the biofilm and be subsequently hydrolyzed and biodegraded. Drury et al. (1993) found that micro beads added in the bulk liquid moved inside the biofilm. These findings suggest that particulate substrates can be directly transported inside the biofilm. Therefore, one should consider both direct and indirect transport of the particulate substrate when modeling the process (Ro, 1995).

3.5 Description of Expected Mathematical Models For Biofilters

There have been many attempts to formulate mathematically the degree of purification achieved by a biological filter. However, analysis of the process is very difficult and the reliability of prediction tends to be uncertain because of the many variables involved, including the unstable characteristics of the hydraulic film and biofilm phases. A universal design equation is still not available because of factors such as variation in the treatability of sewage and the decline in treatability as treatment proceeds.

In the study, the medium volume, backwashing frequency, and recycling flowrate will be used as the critical variables in order that the observed removal of BOD within a biofilter could be described in terms of first-order kinetics.

In sewage biological treatment terms, biofilm kinetics for biofilters is a mathematical method of describing the overall performance of a process in terms of rate of change. Reactions are described as zero-order or first-order depending on whether rates of reaction are independent or proportional to the concentration of the reactant under consideration. It has been found that the carbonaceous oxidation of organic matter within a biological filter can be described in a simplified way by means of a first-order reaction. This is to infer that at any time the rate of change in concentration of BOD is proportional to the remaining concentration of BOD present, i.e.

The exponential law of growth:

$$\frac{dS}{dt} = -k_1 S \tag{3.18}$$

If setting S_I and S_E as the initial BOD and final BOD for the biofilter, mg/L, the removal ratio expression is

$$\frac{S_E}{S_I} = e^{-k_1 t_f} \tag{3.19}$$

where t_f is the time for substrate to travel through the biofilter.

Thus, the observed fraction of BOD remaining in an effluent has been related directly to residence time within the biofilter. Another way is to apply hydraulic loading (influent flowrate) and recycling flowrate to express the removal ratio of BOD for the biofilter.

$$\frac{S_E}{S_I} = e^{-K_r A_S / q_R} \tag{3.20}$$

where A_S is the specific surface area (SSA) for support bead media, m^2/m^3 bead; q_R is the hydraulic loading (influent flowrate), m^3/m^3(bead)-day; K_r is the reaction rate constant expressed in m/d thus rendering the expression dimensionless.

CHAPTER 4: DEVELOPMENT OF MATHEMATICAL MODEL

4.1 Multi-Substrate Biofilm (MSB) Model Formation

A multi-substrate biofilm model (MSB) is developed based on the Monod expression for assisting in the design and operation of middle- and small-size sewage treatment works. The model is used for calculating effluent concentrations, BOD removal rates, OCF, bead bed volumes, and hydraulic loading.

The FBF biofilter system consists of a settling tank, aeration system, bead bed, backwashing system, recycling system, and influent distribution area. Figure 4.1 illustrates the envisioned treatment configuration. The sewage influent comes from the overflow of a raw wastewater septic tank. It enters into biofilter from the bottom of bead bed along with the recycling influent. The recycling effluent comes out of the top of biofilter and forms a partial effluent of system, which is discharged. Air provided by the pump goes into the effluent standing pipe. To initiate this model simulation, the following assumptions are necessary.

(1) The amount of oxygen in biofilter system is consumed only when organic matter is oxidized in the filter, oxygen consumption is reduced compared to BOD removal by backwash water removal;

(2) Particulate and soluble BOD are assumed to degrade equally, physical removal processes are neglected;

(3) Nitrification will not be considered in this model;

(4) The substrate and DO are uniform and the completely mixed assumption applies anywhere in the biofilter due to the recirculating mode;

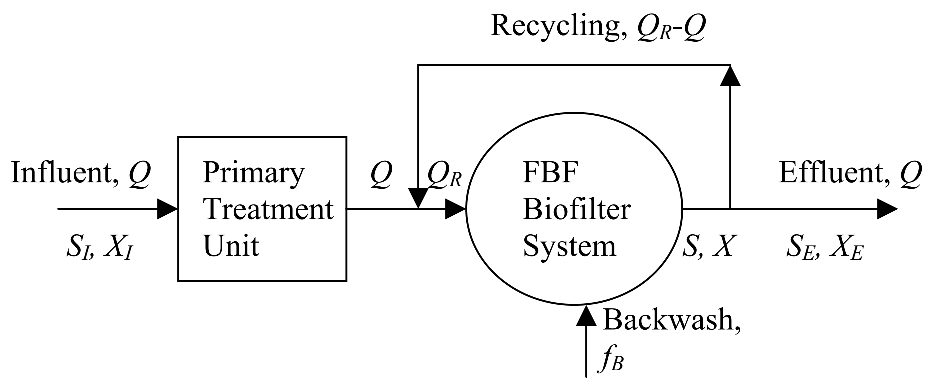

Figure 4.1 The Envisioned Treatment System Configuration with FBF Biofilter

(5) No wastewater enters the biofilter system except the system influent from the settling tank overflow;

4.2 Parameters Selection and Determination

For the model formation, coefficient selection and calibration are very important. The following discussion is about parameters determination and their calculation for the model applications.

4.2.1 Determination of Parameters

For heterotrophs, maximum growth rate constant, μ_m, was suggested to be 4.8 d^{-1} (Wanner and Gujer, 1986). For autotrophs, it was 0.95 d^{-1}. However, a higher μ_m was suggested to be 1.32 d^{-1} (McCarty and Lawrence, 1970).

For the yield coefficient, Y, it was suggested to be the range from 0.24 to 0.64 based on the glucose usage (Eckenfelder, 1975). Wanner and Gujer (1986) suggested an average value of 0.40. The value of cell mass concentration, X_B, is shown to be dependent on total thickness of biofilm and the amount of shear force acting on the biofilm layer (Rittmann and McCarty, 1980). Hoehn and Ray (1980) reported values in the biofilms ranging from 20 to 50 mg/cm^3. The diffusivity of organic substrate was suggested as 0.83 cm^2/d and that of DO as 1.75 cm^2/d by Wanner and Gujer (1986).

Sensitivity of different kinetic parameters is observed by using a range of values for each kinetic parameter. The range of values considered for various kinetic parameters are taken from the research studies mentioned above. For the comparison of parameters, this study will compare the kinetic model parameters for FBF system to that of several previous researchers' model.

4.2.2 Key Parameters

The oxygen consumed in filtration (OCF, kg/m^3-day) is a measure of the rate of total oxygen consumption within the biofilter.

$$OCF = \frac{Q_R(C_I - C_E)}{V_B} \tag{4.1}$$

where C_I is the DO concentration of the influent to the bead bed, mg O_2/L; C_E is the DO concentration of the effluent from the bead bed, mg O_2/L; Q_R is the flowrate through bead bed, m^3/day; V_B is bead bed volume, m^3.

BOD loading (L_{BOD}, kg/m^3-day) is an important organic loading parameter.

$$L_{BOD} = \frac{Q(S_I - S_E)}{V_B} \tag{4.2}$$

where S_I is BOD influent concentration of the biofilter system, mg/L; S_E is BOD effluent concentration of the biofilter system, mg/L; Q is the daily hydraulic loading, m^3/day.

The following mass balance expression is presented to calculate bead bed volume, V_B, under the completely mixed assumption:

$$\frac{dS}{dt}V_B = \frac{\mu_{max}}{Y} \cdot \theta^{(T-20)} \cdot x_B \cdot V_B \cdot \frac{S_E}{K_S + S_E} + Q \cdot (S_I - S_E) \tag{4.3}$$

Where θ is temperature coefficient, typically is 1.03~1.094 (Leslie Grady *et al.*, 1999); T is temperature in ^oC; and x_B is biomass rate, kg/m^3.

When the system is at steady state, $\frac{dS}{dt}V_B = 0$. And, assuming

$$K_{max} = \frac{\mu_{max}}{Y} \cdot x_B \tag{4.4}$$

Then,

$$Q(S_I - S_E) = K_{\max} \cdot \theta^{(T-20)} \cdot V_B \cdot \frac{S_E}{K_S + S_E} \tag{4.5}$$

and finally isolating V_B:

$$V_B = \frac{Q(S_I - S_E)}{K_{\max} \cdot \theta^{(T-20)} \cdot \frac{S_E}{K_S + S_E}} \tag{4.6}$$

Alternatively, Equation 4.8 can be rearranged to:

$$\frac{Q(S_I - S_E)}{V_B} = K_{\max} \cdot \theta^{(T-20)} \cdot \frac{S_E}{K_S + S_E} \tag{4.7}$$

Equation 4.6 can be used to define the bead bed volume under the condition demanded (i.e. Eq. 4.7) that the bed volume normalized load must equal the bed volume normalized conversion capabilities under the given substrate and temperature conditions.

Once the volume of the bed, V_B, is determined by application of Equation 4.6, then the recirculating flow, Q_R, must be defined to assure that the oxygen supply to the bed is sufficient to avoid oxygen limitation or depletion.

The relationship between BOD removal and oxygen consumption is complicated by the effect of backwashing and sludge removal. The bed cleaning process diverts a large portion of the oxygen demand to the sludge handling system as both soluble and particulate BOD is removed but not necessarily oxidized in the bead bed. To address this issue, an empirical correction factor, C_{BW}, is introduced:

$$C_{BW} = \frac{L_{BOD}}{OCF} \tag{4.8}$$

Allowing an approximate linkage between BOD removal and oxygen consumption to be established

The mass balance relationship for oxygen analysis is:

$$\frac{dC}{dt}V_B = Q_R \cdot C_I - Q_R \cdot C_E - \frac{K_{\max}}{C_{BW}} \cdot \theta^{(T-20)} \cdot V_B \cdot \frac{S_E}{K_S + S_E} \tag{4.9}$$

Assuming that an effluent oxygen level, C_E, of 1 mg/L is sufficient to avoid oxygen depletion and that an influent dissolved oxygen concentration equivalent to 60 percent of the oxygen saturation level, C_S in mg/L, under a steady state assumption Equation 4.9 can be rearranged to solve for the mandatory recirculation rate, Q_R:

$$Q_R = \frac{\frac{K_{\max}}{C_{BW}} \cdot \theta^{(T-20)} \cdot V_B \cdot \frac{S_E}{K_S + S_E}}{(0.6 \cdot C_{Sat} - 1.0)} \tag{4.10}$$

The bead bed hydraulic loading, q_R (m^3/m^3 bead bed-day), can be calculated by using the following equation,.

$$q_R = \frac{Q_R}{V_B} \tag{4.11}$$

The system hydraulic loading rate, q_S (m^3/m^3 -day), can be calculated as follows:

$$q_S = \frac{Q}{V_B} \tag{4.12}$$

CHAPTER 5: MODEL CALCULATION AND SIMULATIONS

5.1 MSB Model Case Studies

5.1.1 CASE I: The Relationship between Population Floating Bead Volume

For small treatment systems, their flowrate and wastewater characteristics differ significantly from those of large systems. Table 5.1 shows the typical wastewater flowrate from residential dwellings and small communities (Metcalf & Eddy, 1991).

The Table 5.2 lists parameter ranges derived from ongoing floating bead filter research (Wagener *et al.*, 2002). The given condition for this case is listed Table 5.3. The assumed parameter values in this case are given in Table 5.4 to calculate bead bed volume, recycling flowrate, and hydraulic loading per m^3 bead bed, which are presented in Table 5.5. The relationship of population and bead bed volume is expressed as:

$$Q = fP \tag{5.1}$$

Where f is unit flowrate, Liter/capita-day and P is population, capita.

Table 5.1 Typical Wastewater Flow Rate from Residential Dwellings and Small Communities (Metcalf & Eddy, 1991)

Type of Dwelling	Wastewater Flow Rate			
	Gal/capita-day		Liter/capita-day	
Single family				
Summer	35~50	42	130~190	159
Low income	40~55	45	150~210	170
Median income	40~80	55	150~300	210
Luxury homes	50~100	65	190~380	250
Apartments	35~50	40	130~190	150
Condominiums	35~50	40	130~190	150

Table 5.2 Parameter Value Range Obtained in Experimental Work (Wagener *et al.*, 2002)

Parameter	Unit	Value Range	
Normalized System Flow, q_S	$m^3/\ m^3$-day	5.7~10.6	7.5
Normalized Recycling Flow, q_R	$m^3/\ m^3$-day	310~440	380
Bead Bed Loading, L_{BOD}	KgBOD/m^3-day	1.0~5.0	2.5
Influent BOD Concentration, S_I	Mg/L	100~150	146
System Organic Loading	KgBOD/m^3-day	1~3.5	1.02
Backwash Correction Factor, C_{BW}	Kg BOD/ Kg O_2	1.8~3.0	2.5
Temperature, T	^oC	6~16 (winter)	10
		12~25 (fall/spring)	20
		22~36 (summer)	28

Table 5.3: Given Conditions for Case I

Parameter	Unit	Value	Expression
Population	capita	2000	P
Unit Flowrate	Liter/capita-day	350	f
Total Flowrate	m^3/day	700	$Q=fP$
Influent BOD Conc.	mg/L	146	S_I
Effluent BOD Conc.	Mg/L	10	S_E
Temperature	^oC	25	T

Table 5.4: Coefficient Value Assumed for Analysis

Parameter	Value	Source
K_{Max}	10	Estimated from observation(Wagener et al., 2002)
θ	1.05	Leslie Grady et al., 1999
K_S	20	Estimated from observation(Wagener et al., 2002)
C_{BW}	2.5	Table 5.2
C_S	11.9 at 8 ^oC	Sawyer & McCarty, 1978
	11.3 at 10 ^oC	
	10.2 at 15 ^oC	
	9.2 at 20 ^oC	
	8.4 at 25 ^oC	

Table 5.5: The Calculation Result of V_B, q_R, q_S , Q_R, and L_{BOD} for Case I

Parameter	Unit	Value
Bead Bed Volume, V_B	m^3	22.38
Bead Bed Loading, q_R	m^3/m^3(bead bed)-day	421.22
Organic Loading, L_{BOD}	kgBOD/m^3bed-day	4.25
System Hydraulic Loading, q_S	m^3/m^3-day	31.28
Recycling Flowrate, Q_R	m^3/day	9425.7

Based on the above assumptions and calculation results, the following several points were concluded:

(1) The foregoing assumes a population in the middle of the range for small communities or dwellings (2000 capita), producing domestic sewage of average strength, with a treatment system served by a volume of floating bead medium designed in accordance with V_B= 22.38 m^3. Calculation of parameters was shown in Table 5.5.

(2) When K_S=20 m/L, T=25 °C, and S_I =146 mg/L are given, if the population of small communities increases under $S_E \leq 10$ mg/L, then volume of bead bed, V_B, changes as follows.

Table 5.6: The V_B Under Different Population for C_{BW}=2.5

Population, capita	1000	2000	3000	4000	5000
Bead Bed Volume, V_B, m^3	11.19	22.38	33.57	44.76	55.94

The relationship between population and bead bed volume V_B is shown in Figure 5.1.

(3) It should be also noted that for the same population range, if still keeping C_{BW}=2.5, T=25 °C, S_I =146 mg/L and $S_E \leq 10.0$ mg/L are required, then the bead bed hydraulic loading, q_R, will be obtained from Equation 4.11.

Table 5.7: The q_R Under different Population for L_{BOD}=2.51 kg/m^3-day

Population, capita	1000	2000	3000	4000	5000
System Hydraulic Loading, Q_R, m^3/day	4713	9426	14139	18852	23564

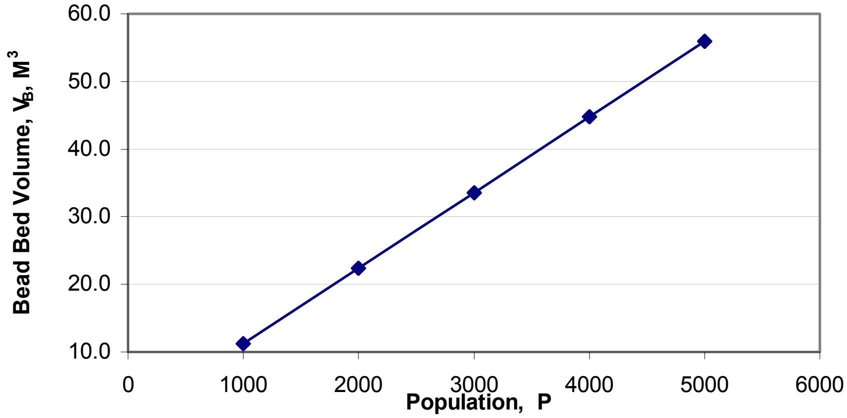

Figure 5.1: **The Relationship between Population, P and Bead Bed Volume V_B**

The relationship between population and bead bed hydraulic loading q_R is shown in Figure 5.2.

5.1.2 CASE II: The Effect of Varying Temperature

For the Case II, the assumptions were given in Table 5.8 to calculate bead bed volume, effluent BOD, and hydraulic loading per m^3 bead bed under different temperatures, which were shown in Table 5.9. The parameter definitions developed for the Case I study are held constant (See Table 5.4).

Using Equations 4.2 to 4.6 to calculate organic loading and volume of bead bed in biofilter system under the different temperature conditions. Let temperatures of sewage in FBF system, T, vary from 6 ~ 16 ^oC for winter, 22 ~ 36 ^oC for summer, and 12~25 ^oC for other seasons in the southern States of US, then calculation result of several parameters in biofilter are listed in Table 5.9.

Based on the above assumptions and results, the following several points were concluded:

(1) Based on the different seasons, variation of temperature does significantly influence the bead bed volume, and OCF. When temperature changes between the range from 6 to 36 ^oC, the bead bed volume decreases 68%. That means that temperature change in different seasons must be considered into the design of a FBF system with effective bead bed volume. On the other hand, temperature also influences the rate of biofilm substrate utilization.

(2) In addition, Temperature effect on bead bed volume V_B and q_R can be seen in Figure 5.3 and 5.4. When weather warm up, say, temperature goes up from 10 ^oC to 28 ^oC, q_R of the bead bed will increase up to 64%.

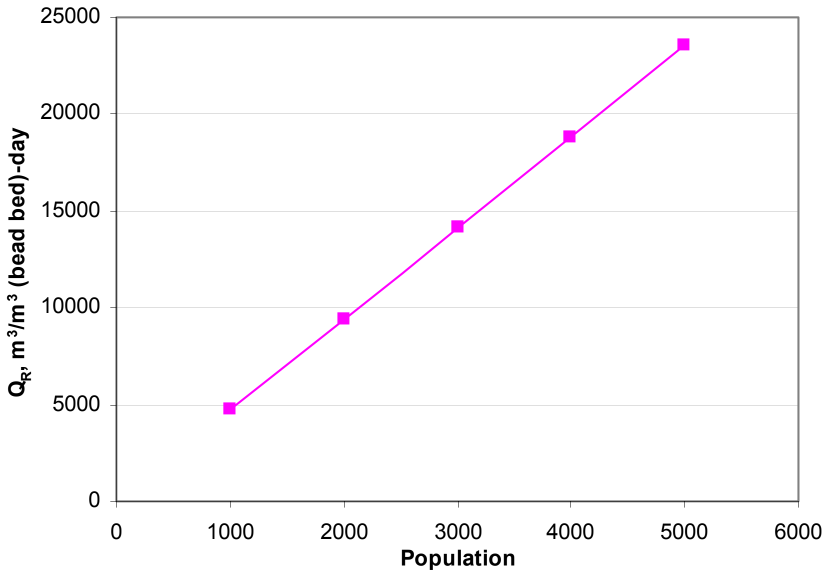

Figure 5.2: **The Relationship between Population, P and System Hydraulic Loading Q_R**

Table 5.8: Given Conditions for Case II

Parameter	Unit	Value	Expression
Population	capita	1000	P
Unit Flowrate	liter/capita-day	350	f
Total Flowrate	m^3/day	350	$Q=fP$
Influent BOD Conc.	mg/L	146	S_I
Effluent BOD Conc.	mg/L	10	S_E
Temperature, T	^oC	6~16(winter)	10
		12~25(others)	20
		22~36(summer)	28

Table 5.9: The Calculation Result of V_B, q_R, q_S, Q_R, OCF, and L_{BOD} Under Different Temperature Condition for Given S_E=10 mg/L

Parameter	Unit	Temperature Value		
		10 ^oC	20 ^oC	28 ^oC
Bead Bed Volume, V_B	m^3	46.5	28.6	19.3
Bead Bed Loading, q_R	m^3/m^3(bead bed)-day	202.6	330.0	487.6
Organic Loading, L_{BOD}	kgBOD/m^3bed-day	2.05	3.33	4.93
System Hydraulic Loading, q_S	m^3/m^3-day	15.05	24.5	36.2
Recycling Flow ate, Q_R	m^3/day	9425.7	9425.7	9425.7
OCF	kgO_2/m^3-day	0.82	1.33	1.97
Effluent BOD, S_E	mg/L	10.0	10.0	10.0

(3) The influence of temperatures in different seasons on the amount of discharged sewage is very significant. Usually, the amount of discharged wastewater in summer is higher than that of other seasons. The unit flowrate (liter/capita-day) is re-considered in the design to a sewage treatment system with FBF biofilter.

(4) The Ratio, $G= V_B(T_1)/V_B(T_2)$, is another parameter which is applicable to the normal operating range of sewage temperatures. It implies that for a given flowrate of sewage influent and in order to maintain a given minimum degree of BOD loading removal, the volume of bead bed in the biofilter required at 10 ^oC would be about 3.0 times that which would be necessary at 30 ^oC.

5.2 MSB Model Summary

For MSB Model, the practical calculation is quite a complicated unless a number of simplifying assumptions are made in the modeling process.

The MSB Model of biofilter performance predicts the composition of bioclarified effluent. It also suggests that its performance is controlled mainly by the hydraulic loading, such as influent flowrate and recycling flowrate, on the effectively wetted surface area (WSA) of the floating bead. Effective WSA depends on the volumetric loading, the type and condition of the floating bead, the frequency of backwash, the condition of recycling system, and the wastewater velocity passed through bead bed. The application of MSB model must assess the effectively wetted surface area and this may be close to the nominal or measured specific surface area if the hydraulic loading is adequate.

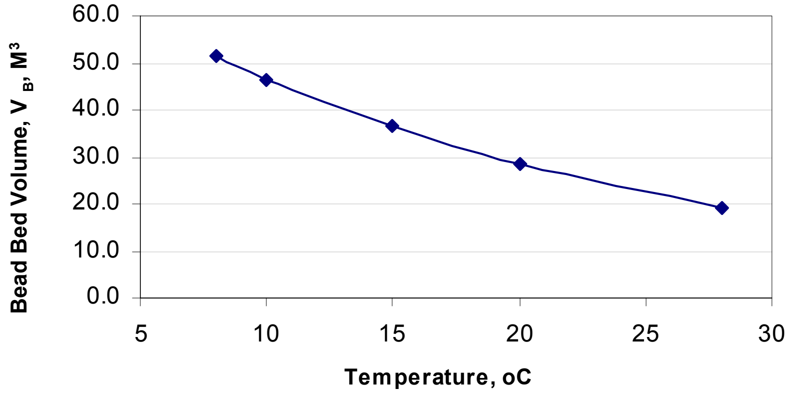

Figure 5.3: Temperature Effect on Bead Bed Volume V_B

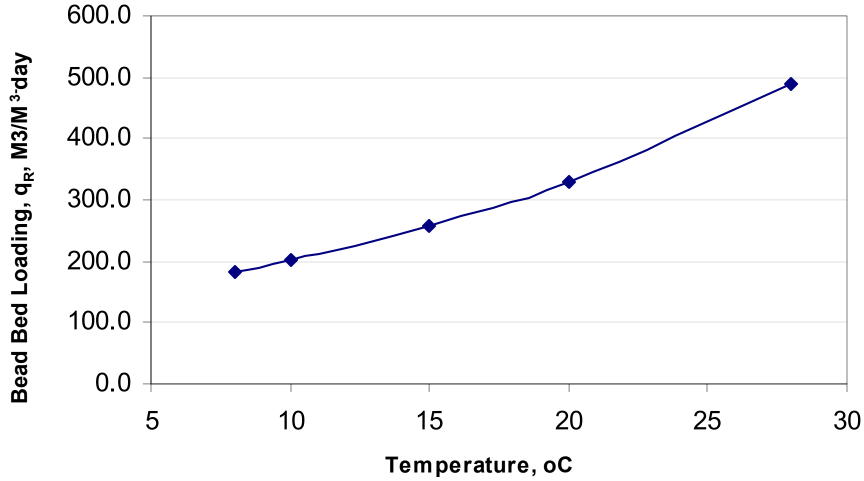

Figure 5.4: Temperature Effect on q_R

At low hydraulic loadings the insensitivity of the MSB model to long-term changes in loading means that a bioclarified effluent BOD of around 10mg/L or less (average) requires relatively large treatment capacity.

A biofilter will be affected by temperature, and those with low and medium loadings also show an effect due to season. Thus performance in winter is worse than in summer. At higher loadings this purely seasonal effect is considerably reduced or absent, although the temperature effect remains. In MSB, both a temperature and seasonal correction will be made when the organic loading is less than a given amount of kg BOD per m^3 each day. The annual average performance will then be seen in the prediction of bioclarified BOD change as time moves on by using MSB. For biofilter with loadings higher than 0.5 kg/m^3-day, this seasonal correction is omitted and the annual average performance occurs at the annual average temperature. Of course, performance of biofilter is not affected by the size of the clarification stage provided that the upward velocity is at maximum flow.

CHAPTER 6: CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

Based on the theoretical description, parameter calibration from experimental data, and model simulation calculations, the following conclusions are drawn from this study.

1. The FBF biofilter system in this study can be applied to a domestic sewage streams treatment plant. The three traditional computer models, zero-order equation, first-order equation, and Monod equation, can effectively describe the biofilm growth, the substrate utilization, and DO transport in FBF system.

2. The Multi-species & Multi-substrate Sewage Operation Model (MSB), was developed and calibrated, and simulated based on the derivation of mathematical methods and the calibration of experimental data. The MSB can approximately provide a computer method for the future design of sewage and other domestic wastewater treatment plants. It can project the bead bed volume and recycling flow required in a FBF application to wastewater treatment.

3. Temperature effect on the biofilter system is seen to be significant. Bead bed volume and hydraulic loading will be increased 68% and 64% when environmental temperature varies with seasons.

6.2 Recommendations

This study provides a framework for the Floating Bead Biofilter processes involved in domestic sewage treatment systems. The developed MSB, a computer model, will be useful in understanding and determining the comprehensive relationships among many bulk operating and design parameters for the sewage treatment systems. It is also limited by the availability of the accurate values of kinetic parameters, properties of biofilm, characteristics of sewage stream, weather, location of treatment plant, and even economic factors. Hence, it is recommended that this computing model, its derived models, and parameters related to these models can be used for preliminary prediction of some key design and operation parameters. Furthermore, the models need to be systematically improved and perfected for future applications. Although, one set of experimental profile data was used in the calibration and simulation of this model, more high-profile data should be used for the development of this model.

REFERENCES

Andrews, J. F., A Mathematical Model for the Continuous Culture of Microorganisms Utilizing Inhibitory Substrates, Biotechnology and Bioengineering, Vol.10, 707-723, 1968

Andrews, J. F., Kinetic Model of Biological Waste Treatment, Biotechnology and Bioengineering Symposium 2, pp. 5-33, 1971

Arvin, E. and Harremoes, P. Concepts and models for biofilm reactor performance, Water Science Technology, Vol.22, p.171, 1990.

Bower, E. J., Theoretical Investigation of Particle Deposition in Biofilm Systems, Water Research, Vol.21, p.1489, 1987.

Characklis, W. G. and Marshall, K. A., Biofilms, 800 pp., John Wiley & Sons, New York, 1990.

Characklis, W. G. and Stewart, P. S., Transport of 1-μM latex particles in Pseudomonas Aeruginosa Biofilms, Biotechnology and Bioengineering, Vol.42, p.167, 1982.

Chiu, S. Y., Erickson, L. E., Fan, L. T., and Kao, I. C., Kinetic Model Identification in Mixed Populations Using Continuous Culture Data, Biotechnology and Bioengineering, Vol. 14, 207-231, 1972.

Chiu, S. Y., Fan, L. T., Kao, I. C., and Erickson, L. E., Kinetic Behavior of Mixed Populations of Activated Sludge, Biotechnology and Bioengineering, Vol.14, 179-199, 1972.

Chung, P. C., Kwak, D. H., and Kim, Y., Application of A Simple Empiric Fixed-film Model to the Fluidized Biofilter Process, Water Science and Technology, (G.B.), 34, 3-4, 427, 1996.

Cooley, P. E., Nitrification of Fish-hatchery Reuse Water Utilizing Low-Density Polyethylene Bead as a Fixed-Film Media Type, M.S. Thesis, University of Idaho, 1979.

Drury, W. J., Characklis, W. G., and Stewart, P. S., Interactions of 1 Mu-M Latex-Particles with Pseudomonas-Aeruginosa Biofilms, Water Research, Vol. 27 (7), 1119-1126, 1993

Eckenfelder, W. W., Commentary to Position Paper by R. E. McKinney, The Value And Use Of Mathematical Models for Activated Sludge Systems, Prog. Water Technology, 7, 35, 1975.

Eckhoff, D. W. and Jenkins, D., Activated Sludge Systems, Kinetics of the Study and Transient States, Report No. 67-12 of the Sanitary Engineering Research Laboratory, University of California, Berkeley, 1967.

Gaudy Jr., A. F. and Gaudy, E. T., Biological Concepts for Design and Operation of the Activated Sludge Process, Environmental Protection Agency Water Pollution Research Series, Report No. 17090-FQJ-09/71, September 1971.

Ghosh, S. and Pohland, F. G., Population Dynamics in Continuous Cultures of Heterogeneous Microbial Populations, Developments in Industrial Microbiology, Vol.12, 295-311, 1971.

Golla, P. S. and Overcamp, T. J., Simple Solutions for Steady State Biofilm Reactors, ASCE Journal of Environmental Engineering, Vol. 116, p.829, 1990.

Goncalves, R. F. and De Oliveria, F. F., Improving the Effluent Quality of Facultative Stabilization Ponds by Means of Submerged Aerated Biofilters, Water Science and Technology, (G.B.) 33, 3, 145, 1996.

Gujer, W. and Wanner, O., Modeling Mixed Population biofilms, In Biofilm, Characklis, W. G. and Marshall, K. C., eds. Wiley, New York, p.397-443, 1990.

Haldane, J. B. S., Enzymes, Longmans, London, 1930.

Harremoes, P., Biofilm Kinetics in Water Pollution Microbiology, Vol. p.71-91, 1978.

Henze, M., Leslie Grady Jr, C. P., Gujer, W., Marais, G. V. R., and Matsuo, T., A General Model for Single-Sludge Wastewater Treatment Systems, Water Resources, Vol.21, No.5, p.505-515, 1987.

Henze, M., Harremoes, P., La Cour Jansen, J., and Arvin, E., Wastewater Treatment: Biological and Chemical Processes, 2nd Edition, Springer, 1997.

Kissel, J. C., McCarty, M., and Street, R. L., Numerical Simulation of Mixed-Culture Biofilm, Journal of Environmental Engineering, Vol.110, No.2, p.393-409, 1984.

Lamb, R. and Owen, G. H., A Suggested Formula for the Process of Biological Filtration, Water Pollution Control, vol. 2, 1970

Leslie Grady, C. P., Jr., Daigger, G. T., and Lim, H. C., Biological Wastewater Treatment, 2nd Ed., Marcel Dekker, Inc., New York, 1999

Levine, A. D., Tchobanoglous, G., and Asano, T., Characterization of the Size Distribution of Contaminants in Wastewater Treatment and Reuse Implications, Journal of WPCF, Vol.57 (7), 805-816, 1985

Malone, R. F., Floating Media Biofilter, U.S. Patent No. 5,126,042 June 30, 1992.

Malone, R. F., Floating Media Hourglass Biofilter, U.S. Patent No. 5,232,586,3 August 1993.

Malone, R. F., Floating Media Hourglass Biofilter, U.S. Patent No. 5,445,740 August 29, 1995.

Malone, R. F., Chitta, B. S., and Drennan, D. G., Optimizing Nitrification in Bead Filters for Warmwater Recirculating Aquaculture Systems, in: Techniques for Modern Aquaculture, Jaw-Kai Wang (Ed.), American Society of Agricultural engineers, Pp.315-325, 1993.

Malone, R. F. and Beecher, L. E., Use of Floating Bead Filters to Recondition Recirculating Waters in Warmwater Aquaculture Production Systems, Aquacultural Engineering, 22 (2000), Pp.57~73.

McCarty, P. L. and Lawrence, A. W., Unified Basis For Biological Treatment Design And Operation, Proceedings of the ASCE, J.SED, Vol.96 (SA3), p.757-778, 1970.

Metcalf and Eddy, Inc., Wastewater Engineering, Treatment, Disposal, and Reuse, McGraw-Hill, Inc., NY, 1991.

Monod, J., The Growth of Bacterial Cultures, Annual Review of Microbiology, 3:371-394, 1949.

Mouri, M. and Niwa, C., Pilot-Plant Studies On Filtration Of Raw Sewage Using Floating Filter Media And Multiple Filter Column Inlets, Water Science And Technology, Vol. 28 (7), 143-151, 1993

Nicoll, Eric H., Aspects of Small Water Pollution Control Works, J. Institute of Public Health Engineers, Vol.12, 1988.

Ødegaard, H, Small Wastewater Treatment Plants: 3. Selected Proceedings of the 3rd International Specialist Conference on Design and Operation of Small Wastewater Treatment Plants, Malaysia, Water Science and Technology, Vol. 35 (6), R7-R7, 1997

Ødegaard, H. and Helness, H., Floating Filters For Particle Removal In Sewage Treatment, Journal of the Chartered Institution of Water and Environmental Management, Vol. 13 (5), 338-342, 1999

Okubo, T., Okada, M., Murakami, A., and Inamori, Y., Influence of Daily Variation of Flow and Pollution Load on the Performance of Submerged Anaerobic Aerobic Biofilm System, Water Science And Technology, Vol. 22 (3-4), 153-160, 1990

Painter, H. A. and Viney, M., Composition of Domestic Sewage, J. of Biochemical and Microbiological Technology and Engineering, Vol. 1, No. 2, p.143, 1959.

Peyton, B. M. and Characklis, W. G., A Statistical-Analysis Of The Effect Of Substrate Utilization And Shear-Stress On The Kinetics Of Biofilm Detachment, Biotechnology and Bioengineering, Vol.41 (7), 728-735, 1993

Rittmann, B. E. and McCarty, P. L., Model of Steady-State Biofilm Kinetics, Biotechnology and Bioengineering, Vol.22, p.2372, 1980.

Rittmann, B. E., The Effect of Shear Stress on Biofilm Loss Rate, Biotechnology and Bioengineering, Vol.24, p.501-506, 1983.

Rittmann, B. E. and McCarty, P. L., Substrate Flux into Biofilms of any Thickness, Journal of Environmental Engineering Division, Vol.107, p.831-849, 1981.

Rittmann, B. E. and Manem, J. A., Development and Experimental Evaluation of a Steady-State, Multi-Species Biofilm Model, Biotechnology and Bioengineering, Vol.39, p.914-922, 1992.

Ro, K. S., Degradation of Non-Diffusible Organic Matter in Biofilm Reactor, Water Research, Vol.29, p.387, 1995.

Rohold, L and Harremoes, P., Degradation of Non-Diffusible Organic Matter in Biofilm Reactor, Water Research, Vol.15, p.671, 1981.

Sarner, E., Removal of Particulate and Dissolved Organics in Aerobic Fixed-Film Biological Processes, Journal of Water Pollution Control Federation, Vol.58, p.165, 1986.

Sawyer, C. N. and McCarty, P. L., Chemistry for Environmental Engineering, McGraw-Hill Book Company, 3rd Edition, p.407, 1978

Sewell, G., The Numerical Solution of Ordinary and Differential Equations, Academic Press, Inc. Sa Diego, CA, 1988.

Shieh, W. K. and Keenan, J. D., Fluidized Bed Biofilm Reactor for Wastewater Treatment, Advances in Biochemical Engineering and Biotechnology, Vol.33, p56, 1986.

Siegrist, H. and Gujer, W., Mass Transfer Mechanisms in a Heterotrophic Biofilm, Water Research, 19, 1369-1378, 1985.

Skowlund, C. T. and Kirmse, D. W., Simplified Models For Packed-Bed Biofilm Reactors, Biotechnology and Bioengineering, Vol. 33 (2), 164-172, 1989

Skowlund, C. T., Effect Of Biofilm Growth On Steady-State Biofilm Models, Biotechnology and Bioengineering, Vol. 35 (5), 502-510, 1990

Tanaka, H., Uzman, S., and Dunn, I. J., Kinetics of Nitrification Using a Fluidized Sand Bed Reactor with Attached Growth, Biotechnology and Bioengineering, Vol.23, 1683-1702, 1981

US EPA, "Septic Tank Systems for Large Flow Application (EPA 832-F-00-079)", 2000.

US EPA, "Septage Treatment/ Disposal (EPA 832-F-99-068)", 1999.

Wagener, C. A., Bellelo, S. M., and Malone, R. F., Static Low-Density Media Filter for Organic and Solid Removal from Domestic Wastewater, WEFTEC 2002

Wanner, O. and Gujer, W., A Multi-Species Biofilm Model, Biotechnology and Bioengineering, Vol.28, p.314-328, 1986.

Wimberly, D. M., Development and Evaluation of a Low-Density Media Biofiltration Unit for Use in Recirculating Fish Culture System, Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, p.169, 1990.

APPENDIX: NOMENCLATURE

A: Unit area in the biofilm along diffusion direction, m^2
A_F: Cross-section area of bead bed, m^2
A_S: Specific surface area (SSA) for support bead media m^2/m^3
θ : Temperature adjusted coefficient
BOD: Biochemical Oxygen Demand
C_I: DO concentration of the influent to the bead bed, mg O_2/L
C_E: DO concentration of the effluent from the bead bed, mg O_2/L
C_S: Correction coefficient for the biomass growth in the bead bed
C_{BW}: Backwashing correction coefficient
$CBOD$: Carbonaceous Biochemical Oxygen Demand
D: General diffusion coefficient in the biofilm, m^2/day
D_{DO}: Diffusion coefficient of DO in biofilm, m^2/day
F: Combined influence factor
F_{BW}: Backwashing frequency per day, time/day
f_{DO}: Dissolved oxygen saturation factor, dimensionless
G: Ratio of bead bed volume under different temperature, dimensionless
H_B: Height of bead bed, m
k: equal to μ_m/Y
k_d: Endogenous decay coefficient, d^{-1}
K_r: Reaction scale constant, m^2/m^3
K_S: Half-saturation coefficient for substrate, g/m^3 or mg/L
K_{Sa}: Half-saturation coefficient for dissolved oxygen, g/m^3 or mg/L
L: Biofilm thickness along transfer direction for steady state, m
L': Biofilm thickness along transfer direction for non-steady state, m
L_{BOD}: BOD organic loading in the system, kg/m^3
M_i: Mass transport rate through the cross section
MX-factor: Maximum ratio of BOD loading to OCF value, dimensionless
m : Coefficient relating to properties of the bead medium in the biofilter
OCF: Volumetric oxygen consumption rate by biofilter
P_S: COD removal rate of septic tank, %
Q: Volume of raw influent (or total influent flowrate) applied to the biofilter, m^3/day
Q_r or Q_R: Volume of total recycling flowrate applied to the biofilter, m^3/day
Q_R: Total recycling flowrate in FBF, m^3/day
q_B: Hydraulic loading for bead bed, $q_B \approx q_H$, m^3/m^3 (bead)-day
q_R: Recirculation hydraulic loading for the biofilter, m^3/m^3-day
q_S: Surface hydraulic loading on bead medium, m^3/m^2(bead)-day
R: Recycling rate per day, Q_r/Q, m^3/day
r_T: Total substrate loading, g/m^3-day
r_V: Substrate volumetric reaction rate (or intrinsic substrate removal rate) in the biofilm, g/m^3-day
$r_{V,S}$: Substrate volumetric reaction rate, g/m^3-day
$r_{V,X}$: Biofilm growth rate, g/m^3-day
S: Substrate concentration outside biofilm, mg/L
S_a: DO concentration consumed in the bead bed, g DO/m^3

S_E: Substrate (BOD) concentration of effluent in biofilter, mg/L
S_i: Substrate concentration on the interface, mg/L
S_I: Substrate (BOD) concentration of influent in biofilter, mg/L
S^{max}: Maximum substrate concentration in Andrews equation, mg/L
T: Average temperature of bead bed in the biofilter, ^{o}C
$\overline{u}$: Rate at which biofilm increase in thickness or biofilm growth rate, ($\overline{u} = \frac{dx}{dt}$)
V_B: Volume of bead bed in the biofilter, m^3
x_B: Biomass for bead bed in the biofilter, g/day
x: Transport distance within biofilm, m
Y: Yield coefficient, g biomass/g substrate mass
Y_m: Maximum yield coefficient, dimensionless
η: BOD removal, %
μ: Specific growth rate coefficient, d^{-1}
μ_m: Maximum specific growth rate coefficient, d^{-1}
μ^{max}: Maximum specific growth rate coefficient in Andrews equation, d^{-1}
v_B: Wastewater velocity passing through bead bed, m/day
τ: Hydraulic retention time (HRT), day

VITA

Qiang Wu was born on April 25, 1964, in Hunan Province, P.R. China. After high school, he entered the Xiang Tan University, China, where he graduated with a Bachelor of Science in chemical and environmental engineering in 1985. In the same year, he moved to Beijing to attend the Beijing Institute of Light Industry and became a full-time graduate student. He graduated with a Master of Science in chemical and environmental engineering in 1988. He has been a full time graduate student at Louisiana State University since Spring 1999, and is presently a candidate for the degree of Master of Science degree in Civil and Environmental Engineering.